Int J Life Cycle Assess (2014) 19:1345–1355
DOI 10.1007/s11367-014-0726-1

WOOD AND OTHER RENEWABLE RESOURCES

Life cycle assessment for emerging materials: case study of a garden bed constructed from lumber produced with three different copper treatments

Michael Tsang · David Meyer · Troy Hawkins · Wesley Ingwersen · Phil Sayre

Received: 22 August 2013 / Accepted: 17 February 2014 / Published online: 8 March 2014

Abstract
Purpose The objective of this research was to evaluate the appropriateness of using life cycle assessment (LCA) for new applications that incorporate emerging materials and involve site-specific scenarios. Cradle-to-grave impacts of copper-treated lumber used in a raised garden bed are assessed to identify key methodological challenges and recommendations applying LCA for such purposes as well as to improve sustainability within this application.
Methods The functional unit is a raised garden bed measuring 6.67 board feet (bf) in volume over a period of 20 years. The garden beds are made from softwood lumber such as southern yellow pine. The two treatment options considered were alkaline copper quaternary and micronized copper quaternary. Ecoinvent 2.2 provided certain life cycle inventory (LCI) data needed for the production of each garden bed, while additional primary and secondary sources were accessed to supplement the LCI.
Results and discussion Primary data were not available for all relevant inventory requirements, as was anticipated, but enough secondary data were gathered to conduct a screening-level LCA on these raised garden bed applications. A notable finding was that elimination of organic solvent could result in a more sustainable lumber treatment product. Conclusions are limited by data availability and key methodological challenges facing LCA and emerging materials.
Conclusions Although important data and methodological challenges facing LCA and emerging materials exist, this LCA captured material and process changes that were important drivers of environmental impacts. LCA methods need to be amended to reflect the properties of emerging materials that determine their fate, transport, and impacts to the environment and health. It is not necessary that all recommendations come to light before LCA is applied in the context of emerging materials. Applications of such materials involve many inputs beyond emerging materials that are already properly assessed by LCA. Therefore, LCA should be used in its current state to enhance the decision-making context for the sustainable development of these applications.

Keywords ACQ · Emerging materials · Life cycle assessment · MCQ · Micronized copper · Treated lumber

Responsible editor: Greg Thoma

Electronic supplementary material The online version of this article (doi:10.1007/s11367-014-0726-1) contains supplementary material, which is available to authorized users.

D. Meyer · T. Hawkins · W. Ingwersen · P. Sayre
United States Environmental Protection Agency, Sustainable Technology Division, 26 W Martin Luther King Drive, Cincinnati, OH 45268, USA

M. Tsang (✉)
Associations of Schools of Public Health Fellow, U.S. EPA, 1300 Pennsylvania Ave, NW Washington, DC 20004, USA
e-mail: mptsang@gmail.com

1 Introduction

1.1 Life cycle assessment

The use and application of life cycle assessment (LCA) is widespread among various industries and organizations for evaluating and improving their operations or decision making. Life cycle in the context of LCA signifies distinct stages of a product's life cycle. This includes raw material extraction, material processing, product manufacture, use, and disposal. This definition of life cycle should not to be confused with the biological referring to stages of an organism's life cycle, nor should LCA be confused with life cycle risk assessment (LCRA; Grieger et al. 2012), which considers ecological or human health risk at each life cycle stage of a chemical. LCA is used for product development, service improvement, and

environmental management decisions and was developed as a way to assess a range of impacts from a wide array of chemicals. The approach of LCA, however, has limitations for assessing impacts from inorganic or emerging material impacts. Issues of fate, transport, and transformation of such materials have been previously addressed (Diamond et al. 2009; Gandhi et al. 2011; Hischier and Walser 2012; Gavankar et al. 2012). For instance, something not fully addressed in current LCA methods is the speciation of inorganic metals which can significantly influence toxicity (Gandhi et al. 2011). Beyond the chemical modeling aspects, the current LCA landscape and environmental parameters may be missing important spatial and temporal characteristics, ultimately overshadowing their roles in the overall impacts.

1.2 Treated lumber

Pressure-treated lumber products are injected with preservatives that lend an antimicrobial advantage, extending the life of the product. Copper and copper-containing compounds have been used as wood preservatives widely over the last century. Some treated lumber products utilize micron and sub-micron copper particles to enhance wood preservation. Chromated copper arsenate (CCA) was the most common preservative used for outdoor residential applications (U.S. EPA 2011), until its use was voluntarily restricted in 2003 due to concerns over toxicity. Two common products have replaced CCA: alkaline copper quaternary (ACQ) and micronized copper quaternary (MCQ). Both exploit the antimicrobial properties of copper for protection against fungal decay and termite attack. However, ACQ contains a preservative that utilizes soluble copper species, while MCQ utilizes a particulate copper species. Matsunaga et al. (2009) identified copper particle sizes ranging from 0.05 to 0.70 μm embedded in commercial lumber products that had been pressure-treated with particulate copper carbonate. In their patent, Leach and Zhang (2010) outline the use of particles ranging from 0.001 to 25 μm, but this study does not independently verify the actual values used in such products. The perceived benefits of using micron to sub-micron particles are a reduction in the corrosiveness to galvanized hardware used in construction. Use of micronized copper represents about 50 % of the treated wood market in North America and approximately US $2.45 billion in sales (Lux Research Inc. 2004; Evans et al. 2008).

The sustainability of copper-based lumber treatment has been evaluated by Bolin and Smith (2011) and Scientific Certification and FFK Ltd. (2009). Bolin and Smith (2011) completed a cradle-to-grave LCA comparing ACQ-treated lumber with a wood–plastic composite (WPC) decking with a functional unit of 1,000 board feet (bf). Their life cycle inventory (LCI) included primary production data from 15 different lumber treaters and captured energy demands and transportation requirements for those facilities. Their results indicated that ACQ performs better than WPC across multiple endpoints. Bolin and Smith (2011) found that certain life cycle stages had a greater influence on some impact categories over others. For instance, lumber production accounts for roughly 61 % of both water use and smog potential impacts, while the use phase contributes 53 % of the eutrophication impact. Their LCA did not include leaching of the treatment solution during the use phase, however. A 2009 LCA by Scientific Certification Systems looked at the cradle-to-gate impacts of ACQ compared with MCQ-treated lumber with a functional unit of 1 billion board feet. Although the use phase and end-of-life considerations were not included, the MCQ treatment demonstrated a 10–81 % reduction in potential environmental impacts compared with ACQ (Scientific Certification and FFK Ltd. 2009).

1.3 Objectives

The objective of this research was to evaluate the appropriateness of using life cycle assessment for applications incorporating emerging materials and involving site-specific scenarios. Cradle-to-grave impacts of copper-treated lumber used in a raised garden bed are assessed to identify key methodological challenges and recommendations applying LCA for such purposes as well as to improve sustainability within this application. The raised garden bed scenario dually obligates the analysis of methodological issues with fate, transport, and toxicity methods as well as site-specific concerns in LCA. This work will build upon existing inventories from Bolin and Smith (2011) and Scientific Certification and FFK Ltd. (2009). Given the rapid growth and widespread use of copper quaternary and the site specificity of a garden bed, the identification of methodological issues associated with assessing this application should be more broadly applicable to a variety of LCA applications involving emerging materials. This research demonstrates the recommendations of both Hischier and Walser (2012) and Gavankar et al. (2012) for conducting a screening-level LCA on such an economically relevant emerging product and goes beyond the Scientific Certification and FFK Ltd. (2009) study in order to complete a cradle-to-grave assessment.

2 Methods

2.1 Goal and scope

This life cycle assessment follows the guidance of ISO14040:2006 and 14044:2006 (International Standard Organization 2006). The goal of this LCA was to evaluate the cradle-to-grave impacts of a raised garden bed using variations of copper-treated lumber (Fig. 1). The functional unit was defined as a raised garden bed measuring 6.67 bf in volume. The garden bed was modeled for use over a 20-year time period. The screening-level LCI was compiled using both

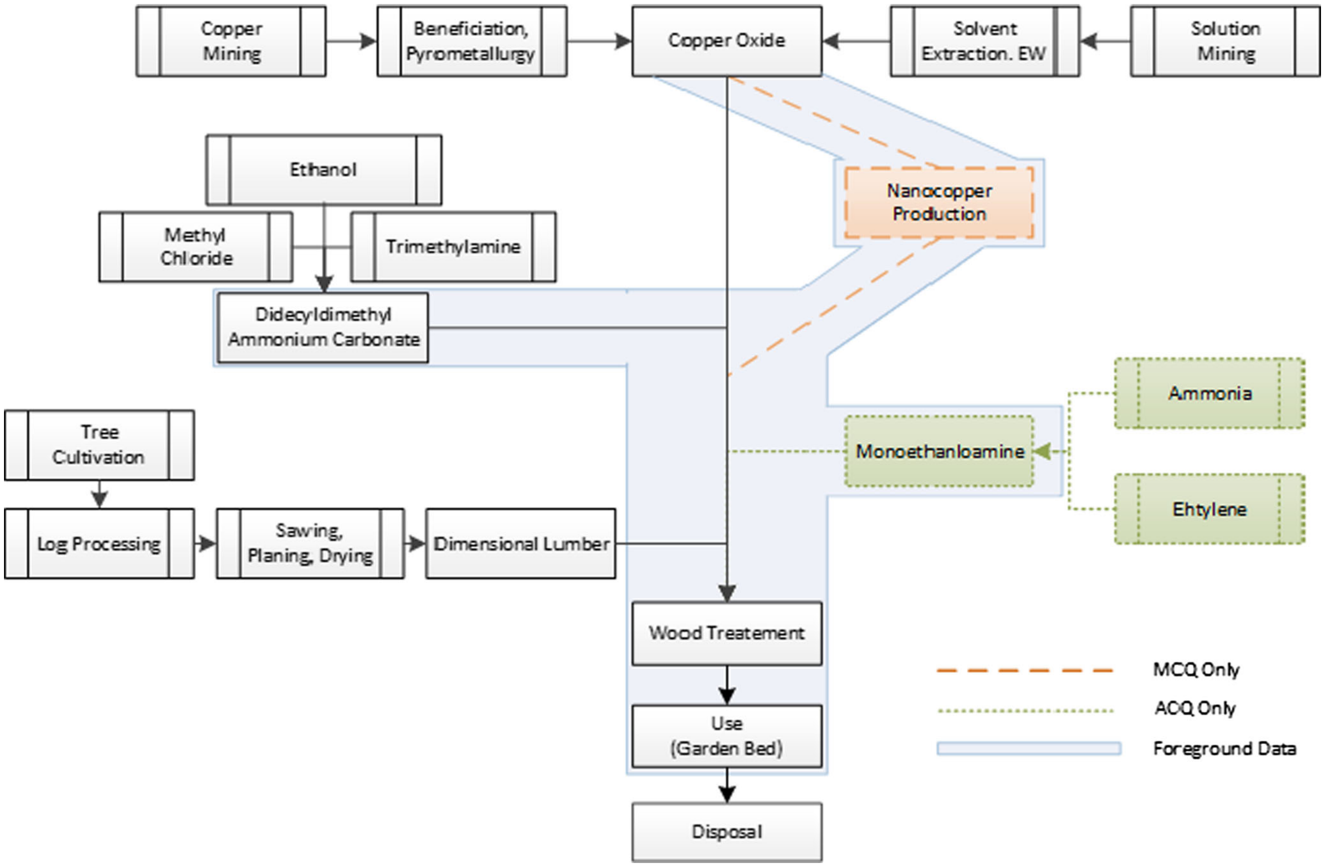

Fig. 1 System boundaries for the main stages throughout the life cycles of ACQ_d and MCQ. Both products utilize all the same processes, except for the *orange* or *green* which are unique to MCQ and ACQ, respectively

primary data and a combination of secondary sources, including patents, scientific literature, and the Ecoinvent 2.2 database.[1]

2.2 Life cycle inventory

2.2.1 Defining ACQ_d, MCQ, and $MCQ_{0.40}$

The frames for all three garden beds were modeled using southern yellow pine lumber. ACQ_d was defined with a retention rate of 0.40 pounds active ingredient per cubic foot (pcf) of lumber, as outlined by the American Wood Protection Association (AWPA) standard UC4A (ground contact, general use; AWPA 2013). MCQ was defined at a retention rate of 0.34 pcf as recommended by the International Code Council's (ICC) ESR-1980 standard (ground contact, general use; ICC 2010). The third lumber product was $MCQ_{0.40}$ and was defined as MCQ, but treated at 0.40 pcf. This third product was hypothetical and was assumed to estimate what portion of the impacts were consequences of solution loading differences (i.e., differences from 0.40 to 0.34 pcf). Cupric oxide (CuO) and didecyldimethylammonium carbonate (DDAC) were chosen as the active ingredients (Leach and Zhang 2010; Great Southern Wood Preserving Inc. 2003; Georgia-Pacific Treated Lumber LLC 2009; Osmose 2010a, b). ACQ_d requires the additional consumption of the non-active ingredient monoethanolamine, which acts as a solvent. Additional colorants and UV inhibitors can be added to the treatment of lumber, but these materials are not standardized and vary based on the producers' specifications. It is assumed that the potential impact from such materials is negligible because they are generally utilized at less than 1 % of the total product weight. Without reliable information on this, the inclusion of these materials was not considered for any product studied in this LCA.

2.2.2 Feedstock production of treatment solutions

Both ACQ and MCQ inventories rely on copper oxide (CuO) background data found in Ecoinvent which includes material, energy, and transportation requirements. The resulting CuO is assumed to be equivalent to that used in both the ACQ treatment solution and also the preparation of MCQ micronized copper oxide. Additional foreground data for MCQ are

[1] Ecoinvent inventory entries often contained European averaged data and was used as surrogate data in lieu of actual US facility data. Such conditions were assumed valid for this study as it provides some consistency with existing LCA studies.

produced by including the processing of CuO into its micronized form. This includes the material, energy, and transportation requirements beyond what was provided for the Ecoinvent CuO process. MCQ requires a particulate CuO that is smaller than the bulk powders used in ACQ_d. Primary production data and energy requirements for producing micronized CuO were gathered from Union Process (Union Processing Inc. 2012) (see Electronic supplementary material 1 for these and all other input calculations). These data are specific for producing CuO with an average size of 1.2 μm. Patents provided the appropriate material inputs and energy requirements necessary for the production of DDAC (Walker 1996; Giede and Rutzen 1981). Monoethanolamine data were taken from Ecoinvent. For transportation requirements, feedstock origins and destinations were modeled inside China, except for micronized copper whose destination was modeled in Akron, Ohio (see Electronic supplementary material). Facilities and capital equipment were not included in the boundaries of this study (U.S. EPA 1994; U.S. Congressional Office of Technology Assessment 1988).

2.2.3 Treatment solution production

Treatment standards for ACQ_d and MCQ lumber demand a 2:1 content ratio of copper to co-biocide[2] (AWPA 2011). This requires that 66.7 % of the solution be represented as CuO and 33.3 % as DDAC. Preservative solutions are generally produced as a mixture of 1 % active ingredient by weight. The remaining 99 % of solution was modeled as water and neglected the optional use of non-active stabilizers, colorants, and biocides (see Electronic supplementary material for calculations). Mixing the components of the preservative solution was assumed as the only energy-consuming step during this phase. Energy requirements for a continuous stir tank reactor provided estimates for electricity usage (see Electronic supplementary material).

2.2.4 Lumber treatment process

Production of untreated dimensional lumber was modeled using the Ecoinvent dataset "*softwood dimensional lumber*." Treatment of ACQ_d lumber requires an active ingredient retention rate of 0.40 pcf (AWPA 2011). This is equivalent to 15.12 kg of active ingredients per 1,000 bf (1 Mbf) of lumber. At a 1 % active ingredient solution, this demands 1,512 kg of ACQ_d solution. MCQ lumber requires an active ingredient retention of 0.34 pcf (ICC 2010). This is equivalent to 1,323 kg of a 1 % active ingredient MCQ solution. Bolin and Smith (2011) calculated the average amount of energy required to pressure treat 1,000 Mbf of ACQ lumber. They surveyed 15 ACQ treatment facilities and received 12 responses detailing the amount of electricity (in kilowatt-hours) used per million board feet. The weighted average was 12 kWh (43.2 MJ). For transportation of inbound chemicals, feedstocks originated in China, except for micronized copper which originated in Akron, Ohio. Inbound transportation of green lumber and outbound transportation of treated lumber as reported in Bolin and Smith (2011) were used. Transportation and electricity data collected by Bolin and Smith (2011) were not allocated based on the type of lumber product produced and represent aggregated values across multiple lumber types. Pressure treatment processes were assumed to be 100 % conserved with all treatment solutions and water being reused in multiple processes (Bolin and Smith 2011).

2.2.5 Use of garden bed

The use phase consisted of transportation to the site of installation and environmental releases (i.e., leaching) from the treated lumber product in situ. Freeman and McIntyre (2008), Cho et al. (2009), and Wang and Kamden (2011) were referenced to supply leaching data for both ACQ_d and MCQ. Leaching methods were either the AWPA E11 standard or those analogous to it and provide a worst-case scenario for leaching potential. These were short-term (14 days), water-based leaching tests on small treated southern pine lumber samples (0.78 in.^{3}) which maximize the exposed surface area. All authors reported the percentage of copper and DDAC leached from lumber samples, except for Wang and Kamden (2011) who did not include DDAC as part of their study. These studies do not specify the species of copper leached, however we assume the copper leached during the use-phase is ionic copper. This should be a reasonable assumption given that the efficacy of particulate copper to protect against wood rot and termite attach depends on its ability to solubilize and travel in its ionic form to deeper parts of the wood (Matsunaga et al. 2009). Additionally, micron and sub-micron metal toxicity can be explained at least in part by its dissolution (Gao et al. 2009; Griffitt et al. 2007, 2008; Franklin et al. 2007). This soluble fraction can be assessed with current life cycle impact assessment (LCIA) methods using the characterization factors (CF) for cupric ion. The ED_{50} value used to create the human health non-carcinogenic CF in USEtox is 54.8 kg per person per lifetime (USEtox 2012). This value was derived from a NOEL reported for canine chronic toxicity test using cupric ion. The EC_{50} value used to create the ecotoxicity CF is 0.11 mg/L. This value was based on the geometric mean of the chronic values reported for 458 aquatic species tested exposed to cupric ion. These ED_{50}/EC_{50} values are used to produce an effect factor that reports the number of cases per kilogram of substance. This CF is labeled an "interim" characterization by USEtox™ authors (USEtox 2012), and impacts dominated by

[2] Actual formulations may vary, but are generally represented in that proportion.

such substances (i.e., use phase) should be interpreted with a lower level of confidence (Rosenbaum et al. 2008). In agreement with the authors of USEtox^{TM}, given this uncertainty, it is still best to incorporate these impacts as, otherwise, there would be neglect to any impact (Rosenbaum et al. 2008; USEtox 2012).

Although long-term soil leaching comparisons were not available in the scientific literature for MCQ and ACQ_d, Lebow (1996) reported results from a select 12-week soil test (AWPA E20) 19×8×200-mm ACQ_d samples resulting in the loss of 15.4 % copper and 12.9 % DDAC. These results are in agreement with the leaching values averaged from the aforementioned leaching data for ACQ_d (Table 1). Both the short-term and longer-term tests were set up in a way that maximizes the surface area-to-volume ratio compared to general treated lumber applications. Therefore, the resulting impacts presented a worst-case scenario.

Bolin and Smith (2011) reported a weighted average "total outbound treated lumber" truck transport from the treatment facility to site of use. This value did not distinguish how the transport is partitioned (i.e., producer to retail center), but it was assumed to include the travel requirements of the producer to the customer. Energy usage was assumed to consist of mostly human labor and insignificant amounts of electricity usage (i.e., electric power tools), which were ignored in the inventory.

2.2.6 Disposal

Disposal of all treated lumber was modeled using a landfill scenario. Data on leaching of copper from MCQ lumber in the landfill were not available in the scientific literature. Townsend and Dubey (2010) did not measure detectable limits of copper in the leachate in their simulation of ACQ-treated lumber in a construction and demolition landfill. They attributed this to the fact that copper readily precipitates out as copper sulfide in strongly reducing sulfide-rich environments (Townsend and Dubey 2010). It is not known how MCQ-treated lumber and its associated micronized copper will behave in the landfill and is an area of needed research. Based on the work of Townsend and Dubey (2010), it was assumed that emissions to groundwater were negligible. This warranted the use of the Ecoinvent process for the disposal of untreated wood to a sanitary landfill. Relevant transportation and energy requirements were captured in this process. This Ecoinvent process also modeled the capture, use, and offsets for methane collection given off from the lumber.

2.3 Impact assessment

The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI), version 2.0 (Bare 2011) was used along with water, fossil, and metal depletion methods from ReCiPe (Goedkoop et al. 2012) and the non-renewable cumulative energy demand method from Ecoinvent (Hischier et al. 2010) to calculate select environmental and human health impacts. Given that the intent of this LCA is to investigate product impacts in the Continental USA, TRACI was used since it is the only methodology tailored specifically to US conditions (Bare 2011). TRACI utilizes midpoint impacts (Table 2), which was viewed as favorable since it reduces the amount of uncertainty in the impact scores (SAIC 2006).

TRACI quantifies human health and ecotoxicity impacts using the USEtox^{TM} model. This model utilizes midpoint impacts, which are preferable as previously mentioned (Rosenbaum et al. 2008). USEtox^{TM} has been vetted by international scientists to create a transparent model. The USEtox^{TM} database is compiled from readily available data sources and values in the scientific literature such as the RIVM e-toxBase, U.S. EPA's ECOTOX, IULCID, and the U.S. EPA IRIS database. The authors of USEtox^{TM} prioritize

Table 1 Sources used to estimate the leaching of copper (cupric ion) and DDAC during the use phase

Reference	Testing method	Materials tested	Retention rate (pcf)	Copper leached (%)	DDAC leached (%)
Cho et al. (2009)	CNS 6716[a]	MCQ–southern pine	0.40	0.49	17.47
	CNS 6716	ACQ–southern pine	0.40	0.69	19.33
Freeman and McIntyre (2008): Michigan State	AWPA E11	MCQ–southern pine	0.40	5.2	7.7
	AWPA E11	ACQ–southern pine	0.40	23.5	10
Freeman and McIntyre (2008): Mississippi State	AWPA E11	MCQ–southern pine	0.40	6.4	8
	AWPA E11	ACQ–southern pine	0.40	19.3	6.4
Wang and Kamden (2011)	AWPA E11	MCQ–southern pine	0.25	3.85	–
Average	–	MCQ	–	3.99	11.06
	–	ACQ	–	14.50	11.91

[a] CNS is a Chinese short-term testing standard similar to AWPA E11

Table 2 TRACI impact categories with additional metal-specific impact categories (italicized) used in our impact assessment

Impact category	Units	Description
Ozone depletion	kg CFC-11[a]	Converts inventory amounts to CFC-11 equivalents
Acidification	H^+	Converts inventory amounts to H^+ equivalents
Smog formation	kg of 0_3	Converts inventory amounts to ozone equivalents
Eutrophication	kg Nitrogen	Converts inventory amounts to nitrogen equivalents
Global warming	kg CO_2	Converts inventory amounts to CO_2 equivalents
Metal depletion	kg Fe	Converts inventory amounts to iron equivalents
Water depletion	m^3	Ratio of quantity of water used versus water reserves
Fossil fuel depletion	kg Oil	Ratio of quantity of water used versus water reserves
Human health criteria	kg PM_{10}[b]	Converts inventory amounts to PM_{10} equivalents
Human health	CTU	Converts LD_{50} values to CTU equivalents
Ecotoxicology	CTU	Converts LC_{50} values to CTU equivalents
Metal human health	*CTU*	*Converts LD_{50} values to CTU equivalents*
Metal ecotoxicology	*CTU*	*Converts LC_{50} values to CTU equivalents*

CTU comparative toxicity units

[a] CFC-11: trichlorofluoromethane

[b] PM_{10}: particulate airborne matter <10 μm in diameter

chronic over acute data, specify a minimum number of species data points, and invoke ED_{50}/EC_{50} values in place of regulatory-based values. The actual toxicity values used depend on the quality and type of the available data and the necessary changes needed to incorporate that data (i.e., use of extrapolation factors).

Given their greater uncertainty, impacts from metals were separated from all other substances for this reason.

2.4 LCA software

OpenLCA is an open-source software program designed for conducting life cycle assessments. Version 1.3.0 was used to perform this assessment. The Ecoinvent 2.2 database and TRACI 2.0 were loaded into OpenLCA prior to running the assessment. Foreground data were imported into the software by converting Excel spreadsheets into relevant Ecospold files using the Ecospold Access Add-in v1.9.17 (Hischier et al. 2010).

3 Results

3.1 Impact assessment results and interpretation

Each garden bed made from ACQ_d resulted in 8–28 % greater impacts compared to one made of MCQ-treated lumber (Fig. 2). Each garden bed made from ACQ_d requires 0.67 kg of monoethanolamine (MEA) per functional unit, which accounts for 0.34–83 % of ACQ_d's greater impacts over MCQ (Table 3; see Electronic supplementary material 2 for a complete list of inputs and outputs to each garden bed). MEA production is dependent on feedstocks of ammonia and ethylene oxide, which are products of natural gas and other fossil fuels. Impact contributions from this were most clearly seen in the fossil fuel depletion impact. Altogether, ACQ_d demanded 1.56 kg more oil equivalents compared with MCQ (Fig. 3), and the production of MEA ultimately accounted for 1.29 kg or 82.7 % of this difference (Table 3). MEA production also contributes more than 50 % of ACQ_d's greater impacts for energy demand, GWP, metal depletion, and acidification, respectively (Table 3). The results for GWP indicate a net *reduction* in global warming potential. Therefore, it should be noted that ACQ has a lower decrease in the global warming potential compared to MCQ and actually possesses a less favorable environmental outcome (Fig. 2). Solution loading (represented as $MCQ_{0.40}$) was an important contributor to ACQ_d's greater impacts (Table 3). Impacts for $MCQ_{0.40}$ range anywhere from 1.9 % greater for energy demand to 12 % larger for ecotoxicity compared with MCQ. Approximately, 17.3–100 % of ACQ_d's greater impacts relative to both MCQ applications can be explained by solution loading (Table 3). Looking again at fossil fuel depletion, it was shown that solution loading accounted for approximately 17.3 % of ACQ_d's greater impact.

For each garden bed scenario, metal ecotoxicity (METP) was almost exclusively driven by copper leaching during the use phase (Fig. 4). Under the assumed conditions (Table 1), leaching of cupric ion during the use phase was estimated to be 35.04, 8.21, and 9.66 % for ACQ_d, MCQ, and $MCQ_{0.40}$, respectively (Table 4). Across the three products, the METP for ACQ_d was 74.5 % greater than MCQ and 70 % more than $MCQ_{0.40}$ (Fig. 4). Leaching of copper during the use phase accounted for a smaller percentage of the metal human toxicity (MHH) compared with METP. ACQ_d's MHH impact was 86.4 % greater than MCQ and 78.9 % more than $MCQ_{0.40}$ (Fig. 4).

Cookson et al. (2010) demonstrate that MCQ performs as least as well as ACQ against rot and termite attack and in many cases performs significantly better (Stirling et al. 2008). Though, any extrapolation into the longevity of each product in the long

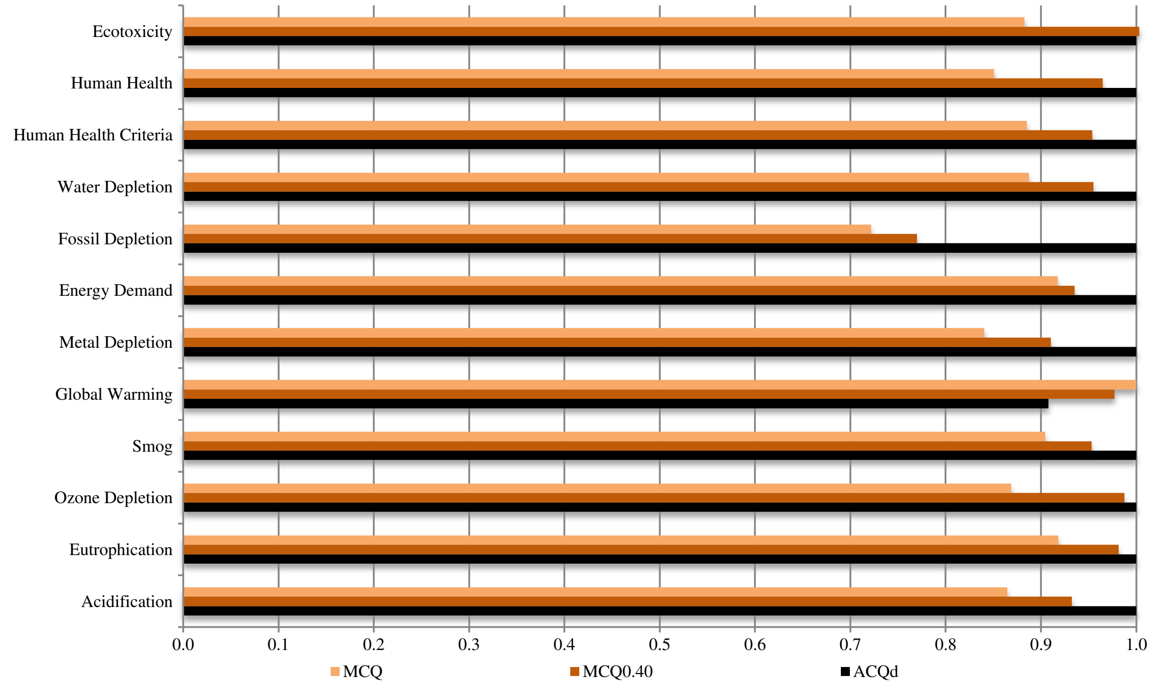

Fig. 2 Normalized impacts with all three products with ACQ_d set as 100 % of the burden in each category (note that for global warming potential, MCQ 0.34 is set at 100 % of the impacts and GWP represents an overall reduction in impact)

run is uncertain. A reasonable assumption could be made that a garden bed made from ACQ may have to be replaced more often than MCQ, but this is not something captured in this LCA.

3.2 Gap analysis: inventory and impact methods

A complete LCI using primary data could not be constructed, as was anticipated, but enough secondary data were gathered (i.e., patents, scientific literature) to populate a screening-level cradle-to-grave inventory for each product. The gaps in primary data can be classified into three general types: (1) accessibility, (2) evaluation, and (3) methodological. The types of data associated with applications involving emerging materials are not different from other typical LCA data and include material input flows, product flows, emissions and waste flows, energy and utility usage, transportation

Table 3 Impact difference between ACQ_d and MCQ: percent attributable to MEA production and solution loading ($MCQ_{0.40}$)

Impact category	ACQ_d−MCQ (difference)	MEA impacts	% of Difference	Solution loading impacts ($MCQ_{0.40}$−MCQ)	% of Difference
Acidification	0.910	0.465	51.15	0.456	50.07
Eutrophication	0.0146	0.00414	28.26	0.0113	77.27
Ozone depletion	2.72E−06	2.66E−07	9.77	2.46E−06	90.43
Smog	0.183	0.0814	44.42	0.0939	51.13
Global warming	3.062	2.29	74.68	0.758	24.83
Metal depletion	13.7	8.05	58.68	6.02	43.85
Energy demand	74.2	59.1	79.58	15.9	21.29
Fossil depletion	1.56	1.29	83.10	0.270	17.30
Water depletion	0.0156	0.00695	44.71	0.00937	60.19
Human health criteria	0.00436	0.00183	42.08	0.00261	59.85
Human health	6.61E−09	1.79E−09	27.06	5.04E−09	76.35
Ecotoxicity	0.432	0.00148	0.34	0.444	>100
Metal human health	1.69E−06	1.67E−07	9.91	5.95E−07	35.26
Metal ecotoxicity	787	1.91	0.24	45.3	5.75

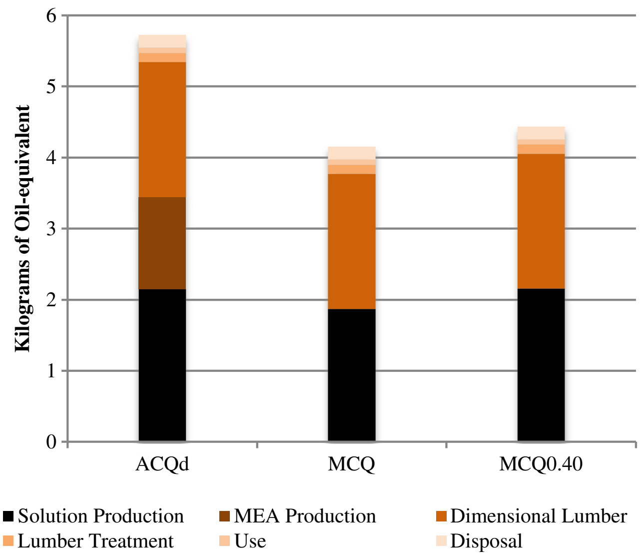

Fig. 3 Fossil fuel depletion for all three treated lumber types in kilograms of oil equivalency

Evaluation gaps are encountered when a life cycle model requires the generation of new datasets through experimentation and field study. For example, the leaching of copper from lumber during use and disposal can be an important contributor to ecological and human health impacts. Proper inclusion of this process in the life cycle model requires extensive product studies to establish the quantity and rate of copper leaching. Furthermore, proper identification and quantification of copper in the leachate were not available in the scientific literature to date. Finally, accurate transport and toxicity data, necessary for the derivation of CFs for micronized copper, do not exist. This last data gap is also relevant for modeling the impacts of micronized copper emissions during manufacturing and processing. The lack of leaching data also creates a gap with regard to the functional (useful) life of the product. The antimicrobial protection will diminish as the copper leaches from the lumber, allowing the lumber to decay and eventually need replacement. The current inventory assumes that both products last the entire life of the garden bed. However, if one product leaches faster than the other, more of that lumber could be required to maintain the functional unit of the study, resulting in larger life cycle impacts.

Methodological gaps can occur when impact assessment models do not accurately reflect the environmental behavior of a chemical or when the necessary model parameters for a chemical do not exist. USEtox™ was first developed to assess organic compounds and relies on many physicochemical parameters such as the octanol–water partition coefficient (K_{ow}) and solubility (Sol_{25}) to describe the transport of these chemicals in the environment. However, these parameters may not be effective transport descriptors for other materials. Resolving this issue has been proven challenging because these values only exist for a small group of inorganic metals

requirements, and infrastructure demands (facility construction, capital equipment, maintenance, etc.) that can be captured using current LCI methods. A portion of the gaps in these datasets arise out of the lack of accessibility created by primary sources. This is especially true for emerging materials because the manufacturing processes are often proprietary in nature and account for a large portion of a company's intellectual capital. Other gaps are related to the exclusion of capital equipment used to produce such materials. For example, this LCA only included the energy demands of the grinding process and did not account for potentially non-trivial maintenance needs such as parts replacement.

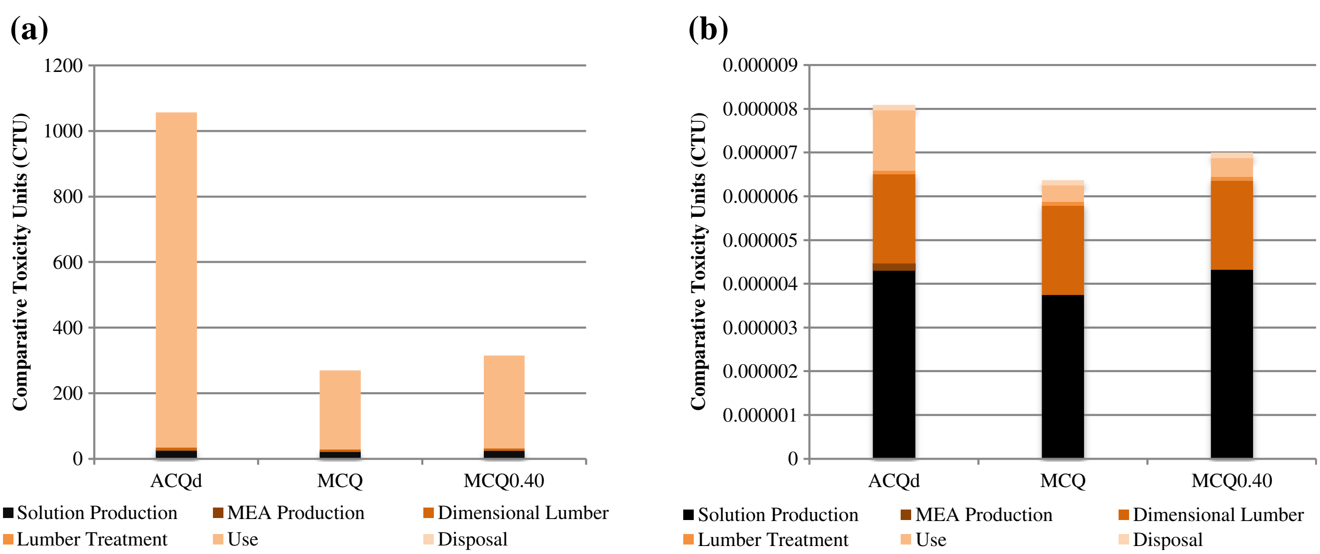

Fig. 4 **a** Impact values for ecotoxicity due to metals (METP) across the life cycle of all three products. **b** Impact values for human health due to metals (MHH) across the life cycle of all three products

Table 4 Amounts of copper leached from each product based on the average leaching assumed from Table 1

Product	Copper leached (mg)	Ecotoxicity (CTU)	Ecotoxicity, metal (CTU)	Human health (CTU)	Human health, metal (CTU)
ACQ_d	35.04	3.69	1,056.67	4.43089E−08	8.00287E−06
MCQ	8.21	3.26	269.31	3.77003E−08	6.31744E−06
$MCQ_{0.40}$	9.66	3.70	314.57	3.77003E−08	6.91209E−06

and are virtually nonexistent for particles or non-traditional materials. In fact, emerging materials may require careful consideration of additional properties which influence their fate and transport. For example, ENM behavior in the environment are often influenced greatly by their surface area, zeta potential, agglomeration/aggregation potential, crystallization, coating, and speciation (Mudunkotuwa et al. 2012; Darlington et al. 2008; Lowry and Casman 2009; Klaine et al. 2008; Hristovski et al. 2011; Jafvert and Kilkarni 2008; Kulkarni and Jafvert 2007). Ultimately, this need to evaluate the applicability of USEtox and adapt it for inorganic substances like copper represents large methodological gaps for the current inventory.

A secondary methodological gap is related to spatial acuteness. The functional unit was chosen with specific interests in mind regarding fate, transport, and potential exposure of lumber treatment chemical consumption of fruits and vegetables grown in a raised garden bed. However, the assessment of impacts to a raised garden bed during its intended use is not captured by the current methodological approach. LCA by design captures national- and global-level impacts which distribute emissions over a greater volume of air, water, and/or soil than would be the case in reality. Local environmental parameters (i.e., pH, salinity, DOC) related to the soil and landscape were similarly not represented. Such values will be important to understanding the transport and exposure of many materials (Hristovski et al. 2011) and may only be truly captured by integrating a more traditional LCRA approach into the LCIA methods. Additional methodological considerations are needed to determine how to report the inputs and outputs of emerging materials with size-dependent properties in the LCI. Current inventory entries are reported on a mass basis, which may not be the appropriate for such materials (Gavankar et al. 2012; Hischier and Walser 2012).

4 Conclusions

The impacts measured from life cycle impact assessments depended on elementary flows linked to environmental compartments which track far-field exposure (i.e., inhalation and ingestion). Although not unique to this LCA alone, impact pathways that involve near-field exposures (i.e., leaching of active ingredients during the use phase) were not captured in these results. In order to address this, elementary flows for direct dermal, direct ingestion, or direct inhalation of such materials need to be incorporated into LCIA methods. The characterization of these impacts and far-field flows should also include fate, transport, and exposure models that diverge from those currently based on organic characteristics. Pending such data and methodological gaps, LCA can continue to be used in its current state to enhance decision making as well as identify hot spots of concern in these types of applications. For hot spots that involve more site-specific exposure pathways, LCA can be coupled with human health risk assessment or ecological risk assessment (Shatkin 2008; Bare 2006) to better understand those potential risks.

5 Disclaimer

The U.S. Environmental Protection Agency (U.S. EPA), through its Office of Research and Development, funded the research described here. This article has been reviewed in accordance with the Agency's peer and administrative review policies and approved for publication. Mention of trade names or commercial products does not constitute an endorsement or recommendation for use by the U.S. EPA. This research was supported in part by an appointment of Michael Tsang as a Fellow in the Chemical Safety for Sustainability National Research Program, administered by the Association of Schools of Public Health (ASPH) through Cooperative Agreement No. X3-83388101 between the ASPH and the U.S. EPA. The research described in this article does not reflect official views of ASPH and lends no official endorsement.

Acknowledgments The present work benefited from the input and guidance from many people at the U.S. EPA as well as Christopher Bolin for supplying certain inventory data.

References

AWPA (2011) Standard for alkaline copper quat type D (ACD-D)
AWPA (2013) AWPA technical information. Retrieved from American Wood Protection Association. http://www.awpa.com/references/homeowner.asp
Bare J (2006) Risk assessment and life-cycle assessment (LCIA) for human health cancerous and noncancerous emissions: integrated and complementary with consistency within the US EPA. Human Ecolog Risk Assess 12(3):493–509

Bare J (2011) TRACI 2.0: The tool for the reduction and assessment of chemical and other environmental impacts 2.0. Clean Technologies and Environmental Policy, 13(5), 687-696

Bolin C, Smith S (2011) Life cycle assessment of ACQ-treated lumber with comparison to wood plastic composite decking. J Clean Prod 19(6–7):620–629

Cho C-L, Lin Y-L, Shen J-Y, Lin L-C (2009) Leachability of commercial ammoniacal copper quat and micronized copper quat used in Taiwan 24(3):183–196

Cookson LJ, Creffield JW, McCarthy KJ, Scown DK (2010) Trials on the efficacy of MCQ in Australia. Forest Prodcuts Society

Darlington T, Neigh A, Spencer M, Nguyen O, Oldensburg S (2008) Nanoparticle characteristics affecting environmental fate and transport through soil. Environ Toxicol Chem 28(6):1191–1199

Diamond M, Gandhi N, Adams W, Atherton J, Bhavsar S, Bulle C, Guinee J (2009) The clearwater consensus: the estimation of metal hazard in freshwater. Int J Life Cycle Assess 15(2):143–147

Evans P, Matsunaga H, Kiguchi M (2008) Large-scale application of nanotechnology for wood protection. Nat Nanotechnol 3:577

Franklin NM, Rogers NJ, Apte SC, Batley GE, Gadd GE, Casey PS (2007) Comparative toxicity of nanoparticulate ZnO, bulk ZnO, and $ZnCl_2$ to a freshwater microalga (*P. subcapitata*)—the importance of particle solubility. Environ Sci Technol 41(24):8484–8490

Freeman M, McIntyre C (2008) A comprehensive review of copper based wood preservatives. Forest Prod J 58(11):6

Gandhi N, Diamond M, Huikibregts MA, Guinee JB, Peijnenburg WJ, van de Meent D (2011) Implications of considering metal bioavailability in estimates of freshwater ecotoxicity—examination of two case studies. Int J Life Cycle Assess 16(8):774–787

Gao J, Youn S, Hovspyan A, Llaneza VL, Wang Y, Bitton G, Bonzongo JC (2009) Dispersion and toxicity of selected manufactured nanomaterials in natural river water samples: effects of water chemical composition. Environ Sci Technol 43(9):3322–3328

Gavankar S, Suh S, Keller A (2012) Life cycle assessment at nanoscale: review and recommendations. Int J Life Cycle Assess 17(3):295–303

Georgia-Pacific Treated Lumber LLC (2009) Material safety data sheet (ACQ pressure treated lumber). Atlanta, Georgia

Giede W, Rutzen L (1981) US patent no. 4,275,235

Goedkoop M, Heijungs R, Huijbregts M, De Schryver A, Strujs J, van Zelm R (2012) A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. ReCiPe 2008. Ministry of Housing, Spatial Planning and Environment, Amersfoort, the Netherlands

Great Southern Wood Preserving Inc. (2003) Material safety data sheet (ACQ treated wood). Abbeville, Alabama

Grieger K, Laurent A, Miseljic M, Christensen F, Baun A, Olsen S (2012) Analysis of current research addressing complementary use of LCA and RA for ENMs—have lessons been learned from previous experience with chemicals. J Nanopart Res 14(7):1

Griffitt R, Weil R, Hyndman K, Denslow N, Powers K, Taylor D, Barber D (2007) Exposure to copper nanoparticles causes gill injury and acute lethality in zebrafish (*Danio rerio*). Environ Sci Technol 41(23):8178–8186

Griffitt R, Luo J, Gao J, Bonzongo JC, Barber DS (2008) Effects of particle composition and species on toxicity of metallic nanomaterials in aquatic organisms. Environ Toxicol Chem 27(9): 1972–1978

Hischier R, Walser T (2012) Life cycle assessment of engineered nanomaterials: state of the art and strategies to overcome existing gaps. Sci Total Environ 425:271–282

Hischier R, Althaus HJ, Bauer C, Busser S, Doka G, Frishknecht R, Simons A (2010) Ecoinvent data v2.2. Ecoinvent Centre Swiss Centre for Life Cycle Inventories, Dubendorf, CH. www.ecoinvent.org

Hristovski KD, Westerhoff PK, Posner JD (2011) Octanol–water distribution of engineered nanomaterials. J Environ Sci Health 46(6): 636–647

ICC (2010) ESR-1980. ICC-ES evaluation report. International Code Council

International Standard Organization (2006) ISO 14044 environmental management—LCA—requirements and guidelines. ISO, Geneva

Jafvert CT, Kilkarni PP (2008) Buckminsterfullerene's (C60) octanol–water partition coefficient (K_{ow}) and aqueous solubility. Environ Sci Technol 42(16):5945–5950

Klaine S, Fernandes T, Handy R, Lyon D, Mahendra S, McLaughlin M, Batley G (2008) Nanomaterials in the environment: behavior, fate, bioavailability and effects. Environ Toxicol Chem 27(9): 1825–1851

Kulkarni PP, Jafvert CT (2007) Solubility of C60 in solvent mixtures. Environ Sci Technol 42(3):845–851

Leach R, Zhang J (2010) US patent no. 7,674,481

Lebow S (1996) Leaching of wood preservative components and their mobility in the environment. United States Department of Agriculture, Washington

Lowry G, Casman E (2009) Nanomaterial transport, transformation and fate in the environment. In: Nanomaterials risks and benefits. Springer, The Netherlands, pp 125–137

Lux Research Inc. (2004) Statement of findings: sizing nanotechnology's value chain. AltAssets. http://www.altassets.com/pdfs/sizingnanotechnologysvaluechain.pdf

Matsunaga H, Kiguchi M, Evans PD (2009) Microdistribution of copper-carbonate and iron oxide nanoparticles in treated wood. J Nanopart Res 11(5):1087–1098

Mudunkotuwa I, Pettibone J, Grassian V (2012) Environmental implications of nanoparticle aging in the processing and fate of copper-based nanomaterials. Environ Sci Technol 46(13): 7001–7010

Osmose (2010a) Material safety data sheet (Naturewood treated wood). Buffalo, New York

Osmose (2010b) Material safety data sheet (Smartsense treated wood). Buffalo, New York

Rosenbaum RK, Bachmann TM et al (2008) USEtox: the UNEP-SETAC toxicity model—recommended characterization factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13(7):532–546

Scientific Applications International Corporation (SAIC) (2006) Life cycle assessment: Principles and practice. Cincinnati: U.S. Environmental Protection Agency Office of Research and Development National Risk Management Research Laboratory

Scientific Certification Systems and FFK Ltd. (2009) life cycle assessment: a comparison of alkaline copper quaternary (ACQ) and micronized copper quaternary (MCQ) lumber treatment chemicals

Shatkin JA (2008) Informing environmental decision making by combining life cycle assessment and risk analysis. J Ind Ecol 12(3): 278–281

Stirling R, Drummond J, Zhang J, Ziobro R (2008) Micro-distribution of micronized copper in southern pine. International Research Group on Wood Protection

Townsend T, Dubey ES (2010) Simulated landfills for assessing the leachate quality impacts from Co-453 disposal of hazardous constituents. State University Systems of Florida, Gainesville

U.S. Congressional Office of Technology Assessment (1988) Copper: technology and competitiveness. U.S. Government Printing Office, Washington

U.S. EPA (1994) Extraction and beneficiation of ores and minerals: copper (volume 4). Washington

U.S. EPA (2011) Chromated copper arsenate. Retrieved from U.S. EPA: 460. http://www.epa.gov/oppad001/reregistration/cca/

Union Processing Inc. (2012). Conversation with Bob Freeman of Union Processing Inc. Production of micronized copper oxide. http://www.unionprocess.com

USEtox (2012) Support of USEtox model. USEtox.org. http://www.usetox.org/Support.aspx

Walker LE (1996) US patent no. 5,523,487

Wang L, Kamden PD (2011) Copper migration from micronized copper preservative treated wood in soil contact. Queenstown, NZ